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Methane emissions from rice across a soil organic matter gradient in Alfisols of Arkansas, USA

Joshua Humphreys [a], Kristofor R. Brye [a,*], Casey Rector [a], Edward E. Gbur [b]

[a] Department of Crop, Soil, and Environmental Sciences, University of Arkansas, Fayetteville, AR 72701, USA
[b] Agricultural Statistics Laboratory, University of Arkansas, Fayetteville, AR, USA

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ABSTRACT

Quantifying greenhouse gas (GHG) emissions in agricultural settings has become critically important in determining the magnitude of agricultural impacts on global climate change. Methane (CH_4) is a leading GHG emitted from rice (*Oryza sativa* L.) production and, since soil organic matter (SOM) serves as a substrate for methanogenesis, understanding the relationship between SOM and CH_4 emissions will be essential for attenuating the excessive release of CH_4 to the atmosphere from rice production. The objective of this field study was to evaluate the effects of SOM on CH_4 emissions from rice grown under a full-season flood across several silt-loam soils in eastern Arkansas. Eight soils were collected from various locations around east-central Arkansas to represent a SOM gradient (22.1 to 51.0 Mg ha^{-1} in the top 10 cm) for this field study. Approximately 0.08 m^3 of soil were placed in plastic bins that were buried in a single bay with the pure-line rice cultivar ('LaKast') transplanted into each bin and grown to harvest maturity under a full-season flood. Season-long, area-scaled CH_4 emissions ranged from 63 to 1521 kg CH_4-C ha^{-1} $season^{-1}$ for in-situ and undisturbed field-plot soil and native prairie soil, respectively, and differed ($P < 0.01$) among the soils evaluated. Rice grown in soil from under a managed grassland, which had the largest SOM content in the top 10 cm, produced the second largest CH_4 emissions (1166 kg CH_4-C ha^{-1} $season^{-1}$). Methane emissions increased linearly ($P < 0.05$) with increasing SOM and total carbon concentration ($R^2 = 0.81$ and 0.85, respectively). Greater understanding of the influence of SOM on CH_4 emissions is essential for assessing GHG impacts from rice production and for refining predictions of CH_4 and total GHG emissions.

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1. Introduction

Anthropogenically induced climate change may be the greatest environmental challenge humans will collectively face in the next 100 years (Intergovernmental Panel on Climate Change (IPCC), 2014a,b). As rainfall patterns change, global temperatures increase, and human populations rise, increasing food production via improved soil health and water resource management will become paramount for sustainable resource use and continued survival (IPCC, 2014a,b). Natural resource management tools are needed in agricultural production to not only increase yield, but reduce climate-change drivers, such as greenhouse gas (GHG) emissions.

The main anthropogenically and naturally produced GHGs are carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4), all of which have experienced increased atmospheric concentrations to unprecedented levels not observed for 800,000 years (IPCC, 2014a,b). A common baseline for GHG concentrations is to use the pre-Industrial Revolution atmospheric GHG concentrations, which were 280 ppm for CO_2, 0.7 ppm for CH_4, and 0.18 to 0.26 ppm for N_2O (Forster et al., 2007). However, by 2005, these same GHGs had mean atmospheric concentrations of 379 ppm for CO_2, 1.8 ppm for CH_4, and 0.32 ppm for N_2O (Forster et al., 2007). Cumulative CO_2 emissions from 1750 to 2012 were 2040 ± 310 Gt CO_2, with ~40% of those emissions remaining in the atmosphere, ~30% being absorbed by the oceans, and the remaining ~30% being sequestered in plants and soils (IPCC, 2014a,b). Total US GHG emissions increased by 8.4% from 1990 to 2010, with a 1.6% decrease from 2010 to 2011 to 108 Tg of CO_2 equivalents (IPCC, 2014a, b). Total GHG emissions peaked in the US during 2007 at 7263 Tg of CO_2 equivalents.

Methane is a potent GHG that is produced in saturated- and/or flooded-soil conditions, due to the absence of oxygen (i.e., anoxic or anaerobic conditions), as a byproduct of chemical carbon (C) reduction (Ferry, 1992). During C reduction, C in soil organic matter (SOM) is converted to CH_4 by methanogens. Methanogens use fermentation products, such as acetic acid and CO_2, which are produced by other soil microbes, as food and an energy source, where CH_4 is produced as a by-product of the reactions. Since agriculture is responsible for 10 to

* Corresponding author at: Department of Crop, Soil, and Environmental Sciences, 115 Plant Sciences Building, University of Arkansas Fayetteville, AR 72701, USA.
E-mail addresses: jjhumphr@email.uark.edu (J. Humphreys), kbrye@uark.edu (K.R. Brye), cr001@email.uark.edu (C. Rector), egbur@uark.edu (E.E. Gbur).

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12% of total global anthropogenic GHG emissions, accounting for nearly 50% of global CH_4 emissions alone (Smith et al., 2007), mitigation of CH_4 production and release in agricultural settings, particularly in areas of concentrated rice (*Oryza sativa* L.) production, will have profound importance for future resource sustainability.

The main agricultural sources of CH_4 emissions in the US are enteric fermentation and manure management, with over 95% of total agriculturally related CH_4 emissions as of 2012 (IPCC, 2014a,b). The natural sources of CH_4 emissions are wetlands (23%), plants (6%), termites (3%), oceans (3%), and gas hydrates (2%) (Conrad, 2009). The anthropogenically influenced sources of CH_4 emissions are ruminants (17%), rice fields (10%), landfills (7%), biomass burning (7%), and sewage treatment (4%), while the remaining 18% is attributed to fossil fuel burning (Conrad, 2009). As of 2011, total CH_4 emissions from rice production represented 1.1% of the total US budget of CH_4 emissions to the atmosphere (IPCC, 2014a,b). However, rice cultivation and residue burning make up 3.7% of the total agricultural CH_4 releases (IPCC, 2014a,b). Between 1990 and 2014, annual CH_4 emissions from rice cultivation varied between 476 and 575 kT (kilotons), whereas CH_4 emissions in 2015 alone represented a 30% decrease compared to 1990 emissions (USEPA, 2017). In 2015, total estimated CH_4 emissions from rice production were 11.2 MMT (million metric tons) of CO_2 equivalents in the US (USEPA, 2017). This substantial amount of GHG production substantiates further examination into mitigation of GHG emissions, particularly for CH_4, from rice production.

The importance of quantifying the impact of CH_4 emissions from rice production cannot be overstated. Rice is the predominant staple food for 17 countries in Asia and the Pacific, nine countries in North and South America, and eight countries in Africa (FAO, 2004). Compared to other cultivated grain crops, rice is unique in that the majority of global rice production, and most of the rice production in the US, occurs under flooded-soil conditions for most of the growing season. In the US, CH_4 emissions from agricultural sources are closely tied to the regional geographic distribution of where rice production occurs, whereas Arkansas, California, Louisiana, and Missouri were the top four rice-producing states in the US in 2015 (NASS, 2016). Based on rice yields, Arkansas produced an estimated 3.8 MMT CO_2 equivalents in 2015 from rice cultivation alone (USEPA, 2017).

Rice grown in the direct-seeded, delayed-flood rice production system common in Arkansas also differs substantially from more traditional rice production systems, where rice is hand-transplanted directly to a flooded field. The direct-seeded, delayed-flood rice production system initiates the flood on a rice field four to six weeks after planting, thus limiting the time the flood is present over the entire growing season, which is unlike the water-seeded or hand-transplanted rice systems where flooded soil conditions persist nearly year-round. In Arkansas, the direct-seeded, delayed-flood rice production system is the dominate system, accounting for ~85% of the total planted-rice area, for which annual irrigation water use averaged 763 mm over a 10-yr period between 2003 and 2012 (Henry et al., 2016). In Arkansas, a unique and main irrigation standard is the use of the multiple-inlet irrigation, which uses poly-tubing as a means of irrigating rice to conserve water and labor (Henry et al., 2016). As of 2015, rice producers utilized multiple-inlet irrigation on ~41% of the rice area in Arkansas (Hardke, 2016). Groundwater is used to irrigate over 74% of the rice area in Arkansas, with the remaining area irrigated with surface water obtained from reservoirs, streams, or bayous (Hardke, 2016).

In the soil environment, whether undisturbed or agricultural, the main source of CH_4 in the soil column is in the topsoil, where >99% of the total soil-produced CH_4 is emitted (Mitra et al., 2002). Due to the flooded-soil nature associated with rice production, CH_4 diffusion through the water column is generally slow. However, the main release mechanism of CH_4 from rice cultivation to the atmosphere from below a column of water has been via passive transport through the aerenchyma tissue of the rice plants themselves (Cicerone and Shetter, 1981; Yu et al., 1997; Dannenburg and Conrad, 1999; Groot et al., 2005). Since soil microbes in an anaerobic setting eventually require C as an electron acceptor to carry out metabolic processes, increasing the supply of SOM would likely increase microbial activity and therefore CH_4 production. However, single or multiple soil property, particularly SOM, correlations with CH_4 emissions have been inconclusive (Wang et al., 1993; Watanabe and Kimura, 1999).

The challenges of population growth require a clear understanding of soil conditions and management practices so that innovative techniques can be developed and implemented to off-set potential negative agronomic and ecological/environmental effects of climate change. The pressure to expand production into previously uncultivated land is tremendous. Previously uncultivated land has the temptations of increased soil fertility leading to greater yields, but also potentially negative environmental drawbacks. One of these potential drawbacks is increased CH_4 production from rice production due to large initial SOM.

Since CH_4 can only be produced if there is a source of reducible C in the soil, it stands to reason that soils with a greater initial SOM concentration would produce greater amounts of CH_4 in the flooded-soil condition associated with rice production (Ferry, 1992). However, to date, this relationship has not been well demonstrated. Therefore, the objective of this field study was to evaluate the effect of SOM on season-long CH_4 emissions from a pure-line cultivar planted in numerous silt-loam soils and grown under a full-season flood in eastern Arkansas. It was hypothesized that CH_4 emissions would vary among soils with differing initial SOM concentrations and, specifically, CH_4 emissions would increase linearly as SOM concentration increased. It was also hypothesized that the emissions intensity (kg CH_4-C (Mg grain)$^{-1}$) would be inversely related to SOM concentration.

2. Materials and methods

2.1. Site description

Field research was conducted in 2016 at the University of Arkansas System Division of Agriculture's Rice Research and Extension Center (RREC) near Stuttgart, AR (34°27′54.5" N, 91°25′8.6" W) and closely followed procedures outlined in Rogers et al. (2014). The RREC is located in a region in east-central AR known as the Grand Prairie, which is part of the Major Land Resource Area 131D, Southern Mississippi River Terraces, within Arkansas County (USDA, 2006). The study area has been managed in a rice-soybean (*Glycine* max L. [Merr.]) rotation, which is a commonly used rotation for rice production in Arkansas, for >25 years. The slope across the study area was approximately 0.2% to facilitate irrigation water application and removal. The regional climate throughout the study area is temperate, with a mean annual air temperature of 17 °C, which ranges from a mean minimum of 12.7 °C in January to a mean maximum of 23.5 °C in July (NOAA, 2015). The mean annual precipitation for the study area is 1350 mm (NOAA, 2015).

2.2. Field treatments and establishment

Field treatments for this study consisted of eight soils collected from various locations from the agricultural region of east-central Arkansas that established a SOM concentration gradient. Two of the eight soils were collected from the University of Arkansas System Division of Agriculture's Pine Tree Research Station (PTRS; 35° 22.1″ N, 90° 55′ 45.2″ W) in St. Francis County near Colt, AR. One soil was from a Calhoun silt loam (fine-silty, mixed, active, thermic Typic Glossaqualfs) under cultivated agriculture (CA) in a rice-soybean rotation (CA-PT). The second soil was collected from a Henry silt loam (coarse-silty, mixed, active, thermic Typic Fragiaqualfs) under Conservation Reserve Program (CRP) managed grassland landuse that had not been used for cultivated agriculture for at least 15 years. The dominant vegetation in the CRP field was big bluestem (*Andropogon gerardi*) and little bluestem (*Schizachyrium scoparium*). Four of the eight soils were collected from

a private farmstead (i.e., the Seidenstricker Farm) (34° 43′ 40.26″ N, 91° 33′ 10.76″ W) north of Stuttgart, AR, where one soil was a DeWitt silt loam (fine, smectitic, thermic Typic Albaqualfs; USDA, 2015) under native tallgrass prairie (NP) landuse, which had been subject to periodic annual burning, while the other three soils were collected from agricultural landuse immediately adjacent to the native prairie that had been under continuous annual cultivation in a rice-wheat (*Triticum aestivum*)-soybean rotation for 30 (CA-30; DeWitt silt loam), 41 [CA-41; Stuttgart silt loam (fine, smectitic, Albaquultic Hapludalfs)], and 59 (CA-59; Stuttgart silt loam) years. The remaining two of eight soils were collected from the RREC, where one soil had been under cultivated agriculture in a rice-soybean rotation for at least 25 years (CA-25; DeWitt silt loam), while the other soil was from a managed grassland (MG; DeWitt silt loam) mix of fescue (*Festuca* spp.) and Bermudagrass (*Cynodon dactylon*) (i.e., a manicured lawn). Table 1 summarizes additional characteristics of the eight soils and the sites from which the soils were collected. In addition to the eight disturbed-soil treatments, a ninth treatment was evaluated, which consisted of in-situ, undisturbed soil from replicated field plots, referred to hereafter as field-plot soil, that was not excavated and placed into plastic bins.

Between 18 March and 6 May 2016, soils were collected from each site. At each site, soil was manually excavated to a depth of ~50 cm. First, the upper ~20 cm of soil were removed and temporarily set aside on a tarp, while the remaining sub-soil, ~20- to 45-cm depth interval, was manually excavated and placed into a 33-cm wide × 60.7-cm long × 42.6-cm deep, high-density, commercially available plastic bin. Once the sub-soil was in place in the plastic bin, which occupied the bottom ~20 cm of the bin, the upper 20 cm of topsoil was placed in the bin on top of the sub-soil to recreate the original soil profile horizon sequence as best as possible. Each of the eight soils collected from the various sites were collected in triplicate for a total of 24 bins.

All soil-containing bins were transported to the RREC and, on 7 May 2016, the bins were randomly placed within two, 5-m wide × 3-m long areas adjacent to one another that were manually excavated to a depth of ~40 cm. Once all 24 bins had been placed in one of the two excavated areas, soil was manually back-filled around the bins to bury them such that the soil level inside the bins was at the approximate level of the surrounding natural soil. After back-filling soil around the bins, the top ~10 cm of the soil surface in each bin was manually disturbed to simulate tillage by breaking up large clods to create a semi-smooth, uniformly appearing, level seed bed into which rice seedlings would be transplanted.

On 20 May 2016, ~10-cm-tall rice seedlings, which had 4 to 5 leaves, from a nearby area, which had been drill-seeded with the pure-line rice cultivar 'LaKast' on 23 April 2016, were manually transplanted 2- to 4-cm deep into two rows 18-cm apart in each bin to match the planting density in the surrounding drill-seeded area, which was approximately 32.0 plants m^{-2}. On 8 June 2016, the transplanted rice plants in the bins were fertilized with a manually broadcast, pre-flood, optimum recommended rate of 117 kg N ha^{-1} as urea (46% N) [i.e., 5.77 g surface-applied urea per bin]. Since the rice seedlings had already been pre-flood fertilized once before transplanting, the pre-flood N application was used to offset the transplant shock to the rice plants.

A levee that had been previously established around the buried-bin area contained the permanent full-season flood that was established immediately after N fertilization on 9 June 2016 and was maintained at a depth of ~10 cm until harvest maturity. Since the floodwater covered all of the buried-bin area, the floodwater was able to move freely among bins. On 27 June 2016, 18 days after flood establishment, the mid-season, split N application of 117 kg N ha^{-1} was manually broadcast-applied [i.e., 5.77 g surface-applied urea per bin] to the floodwater at the beginning of internode elongation. The rice in the bins was provided extra N beyond the typically recommended 45 kg N ha^{-1} as the mid-season application due to the rice being semi-stressed from transplanting.

In addition to the 24 buried bins containing transplanted rice, four field plots were established adjacent to the buried-bin area in the same full-season-flood bay to evaluate the effect of growing transplanted rice in the bins compared to direct, drilled-seeded rice into native soil in typical field plots. Field plots were 1.6-m wide by 5-m long, with nine drill-seeded rice rows with 18-cm row spacing. Field plots were planted with the pure-line rice cultivar 'LaKast' on 23 April 2016. Similar to the transplanted bins, on 8 June 2016, field plots were manually broadcast-fertilized pre-flood at a rate of 117 kg N ha^{-1} as urea. On 27 June 2016, 18 days after flood establishment, 45 kg N ha^{-1} were manually broadcast-applied to the floodwater for the mid-season, split N application. The field plots were not provided with any extra N beyond the typically recommended 45 kg N ha^{-1} as the mid-season application since rice in the field plots was not transplanted. On 23 August 2016, the flood was released from the bay containing the 24 buried bins with transplanted rice and the four field plots to prepare for harvest.

2.3. Soil sample collection, processing, and analyses

Prior to flood establishment, on 28 May 2016, two soil cores, 4.8 cm in diameter, were collected with a core chamber and slide hammer from the top 10 cm in each bin and field plot for soil property analyses. All soil samples were dried at 70 °C for 72 h, crushed, and sieved through a 2-mm mesh screen. Large, visible pieces of organic material were manually removed prior to crushing and sieving, but smaller pieces of organic material that passed through the sieve were retained in the sample as part of the soil. One set of soil samples per bin/plot was used for particle-size analyses using a modified 12-hr hydrometer method (Gee and Or, 2002).

The second set of soil samples was used for soil chemical property determinations. Electrical conductivity (EC) and soil pH were analyzed potentiometrically in a 1:2 (m/v) soil-water suspension. Soil organic matter concentration was determined by weight-loss-on-ignition after 2 h at 360 °C. Inductively coupled, argon-plasma, atomic emissions spectrometry (Spectro Arcos, Spectro Analytical Instruments, Kleve, Germany) was used to determine Mehlich-3 extractable nutrient (i.e., P, K, Ca, Mg, S, Na, Fe, Mn, Zn, and Cu) concentrations using a

Table 1
Summary of landuse, Major Land Resource Area (MLRA) classification, soil series (sub-group taxonomic classification), and unique landuse feature associated with the various soil treatments selected for evaluation in this study.

Landuse (abbreviation)	MLRA	Soil series (Soil Sub-group)	Unique feature
Native prairie (NP)	131D	Dewitt silt loam (Typic Albaqualfs)	Native tall grass prairie
Managed grassland (MG)	131D	Dewitt silt loam (Typic Albaqualfs)	Manicured lawn
Conservation Resource Program (CRP)	134	Henry silt loam (Typic Fragiaqualfs)	>15 years in CRP
Cultivated agriculture (CA-PT)	134	Calhoun silt loam (Typic Glossaqualfs)	>10 years in rice/soybean rotation
Cultivated agriculture (CA-25)	131D	Dewitt silt loam (Typic Albaqualfs)	>25 years in rice/soybean rotation
Cultivated agriculture (CA-30)	131D	Dewitt silt loam (Typic Albaqualfs)	>30 years in rice/soybean rotation
Cultivated agriculture (CA-41)	131D	Stuttgart silt loam (Albaquultic Hapludalfs)	41 years in rice-wheat-soybean rotation
Cultivated agriculture (CA-59)	131D	Stuttgart silt loam (Albaquultic Hapludalfs)	59 years in rice-wheat-soybean rotation
Cultivated agriculture (FP)	131D	Dewitt silt loam (Typic Albaqualfs)	> 25 years in rice/soybean rotation

1:10 soil-mass-to-extractant-volume ratio (Tucker, 1992). Total nitrogen (TN) and total carbon (TC) concentrations were measured by high-temperature combustion with a VarioMax C:N analyzer (Elementar Americas, Inc., Mt. Laurel, NJ). Since all soils did not effervesce upon treatment with dilute hydrochloric acid, all measured TC was assumed to be organic C. Measured TN and TC concentrations were used to calculate soil C:N ratios on a bin-by-bin or plot-by-plot basis. Based on measured sand and clay fractions and SOM concentrations from the top 10 cm of each bin/plot, soil bulk densities on a bin-by-bin and plot-by-plot basis were estimated from generalized multiple regression equations (Saxton et al., 1986). All measured soil concentrations (mg kg^{-1}) were converted to contents (kg or Mg ha^{-1}) using the estimated bulk densities and 10-cm sampling interval.

2.4. Soil oxidation-reduction potential and temperature measurements

Immediately after flooding of the bay containing the 24 buried bins and four field plots began (9 June 2016), soil oxidation-reduction (redox) potential (Eh) sensors (Model S650KD-OR, Sensorex, Garden Grove, CA), with Ag/AgCl reference solution, were installed vertically to a depth of ~7 cm. One Eh sensor was installed adjacent to a gas-sampling-chamber base collar, described below, in each plot and randomly in two of the three bin replications per soil treatment. In addition to the Eh sensors, chromel-constantan thermocouples were installed horizontally in the bulk soil at a depth of ~7 cm in each plot and in the one remaining bin replication per soil treatment. All sensors were connected to a datalogger (CR 1000, Campbell Scientific, Inc., Logan, UT), which was housed in an environmental enclosure, to record soil temperature and soil Eh at 15-minute intervals, while mean data were output every hour. Measured sensor data were collected weekly. Soil Eh values were corrected to the standard hydrogen electrode by adding 199 mV to each field-measured value (Patrick et al., 1996).

For the purposes of data reporting, both soil Eh and temperature data from the hour during gas sample collection on each measurement date were extracted from the continuously recorded data series for all replicate sensors. The individual hourly soil Eh and temperature data from each weekly measurement date were subsequently used for statistical analyses.

2.5. Gas sample collection and analyses

Similar to procedures used by Rogers et al. (2014), after planting/transplanting and before flooding, a wooden boardwalk system was erected throughout the study area to reduce disturbances to the soil and rice plants and allow easier access to the plots/bins during the growing season for gas sample collection and other plot/bin maintenance and access. The boardwalk was constructed out of 5.1-cm thick × 30.5-cm wide × 3.6-m long pressure-treated, wooden planks set upon 20- × 40-cm concrete blocks before chamber base collar placement in the plots/bins. One chamber base collar, 30-cm in diameter × 30-cm tall, was installed to encompass the third and fourth rice rows in each field plot for gas sampling. One base collar for gas sampling was then set into place in the center of each bin encompassing the majority of both manually transplanted rice rows. Base collars were constructed out of 0.6-cm thick, Schedule 40 polyvinyl chloride (PVC) material and beveled to a 45° angle to facilitate installation. Base collars were inserted ~10 cm into the soil so that four 1.25-cm diameter holes 12 cm from the bottom of the base collar were ~1 cm above the soil when properly inserted to facilitate flood-water movement into and out of the base collar.

Vented, non-steady-state, non-flow-through chambers (Livingston and Hutchinson, 1995) made out of 30-cm diameter Schedule 40 PVC were used for gas sample acquisition for the purpose of CH_4 flux determinations (Rogers et al., 2014). To prevent convection currents inside the chambers that would dilute the ambient, headspace air during sampling, the holes in the base collars were plugged with gray butyl-rubber septa (Voigt Global, part# 73828A-RB, Lawrence, KS) during sampling after flood release.

Chamber extensions, 40 and 60 cm in length depending on the height of the rice plants at the time of sampling, were used to accommodate rice growth during the season. Reflective aluminum tape (CS Hyde, Mylar metallized tape, Lake Villa, IL) was used to cover chamber extensions to reduce temperature variations inside the chamber during use. Tire inner tube cross sections were cut to an ~ 10-cm width and taped to the bottom of all the extensions to function as a seal between the base collar and the chamber extensions during gas sampling.

Chamber caps were constructed with 10-cm-tall sections of 30-cm-diameter PVC, with a 5-mm-thick sheet of PVC glued to the top and covered with reflective aluminum tape. Tire inner tube cross sections, ~10-cm wide, were also taped to the bottom of the caps to serve as a seal between the chamber base collar early in the growing season or upper-most extension later in the season. A 15-cm-long piece of 4.5-mm-inside-diameter (id) copper refrigerator tubing was installed into the side of each cap to maintain atmospheric pressure during gas sampling. On the top of the gas-chamber caps, two 12.5-mm-diameter holes were drilled and plugged with gray butyl-rubber septa for syringe and thermometer insertion. To ensure adequate air mixing in the enclosed gas chamber, a 2.5-cm tall × 2.5-cm wide, 9 V-battery-operated, magnetic levitation fan (Sunon Inc., MagLev, Brea, CA) was installed on the underside of the chamber cap and operated for the duration of gas sampling.

The collection of gas samples from the enclosed chambers was achieved using a 20-mL B-D syringe with a removable 0.5-mm diameter × 25-mm long needle (Beckton Dickson and Co., Franklin Lakes, NJ) that was inserted through the gray butyl-rubber septa installed in the chamber cap. After drawing a gas sample from the chamber, the collected sample was immediately injected into a pre-evacuated, 10-mL, crimp-top glass vial (Agilent Technologies, part# 5182-0838, Santa Clara, CA). Gas sampling occurred weekly between flooding and flood release starting 5 d after flooding. On each sample date, gas samples were collected at 20-min intervals for 1 h, after the chamber was capped and sealed (i.e., the 0-, 20-, 40-, and 60-min marks). At the end of the growing season, prior to harvest, gas sampling occurred 1, 5, and 6 d after flood release. Similar to prior studies (Rogers et al., 2013, 2014), all gas sampling started in the morning between 0800 and 0830 h to minimize temperature fluctuations in the chambers and to maintain continuity with previous research.

During each chamber sampling event, 10-cm soil temperature, relative humidity, ambient air temperature, barometric pressure, and the air temperature inside the chamber were measured. At the end of each gas sampling event, the chamber height to the current water level was recorded so that the interior chamber volume could be accurately calculated. Samples of CH_4 gas standards (i.e., 2, 5, 10, 20, and 50 mg L^{-1}) were collected in the field using a 20-mL B-D syringe with a detachable 0.5-mm-diameter × 25-mm-long needle that was immediately injected into a pre-evacuated, 10-mL, crimp-top glass vial. Immediately prior to field sample analyses, CH_4 gas samples from the same five gas standards were also collected in the laboratory.

Using a flame ionization detector (250 °C) equipped with a gas chromatograph (Model 6890-N; Agilent Technologies, Santa Clara, CA), with a 0.53-mm diameter × 30-m HP-Plot-Q capillary column (Agilent Technologies), gas samples were analyzed for CH_4 concentrations within 48 h of collection. Based on procedures described by Rogers et al. (2014), CH_4 fluxes were calculated by linear regression according to changes in concentrations in the chamber headspace over the 60-min sampling interval. To determine the change in concentration over time, measured concentrations (mL L^{-1}; y axis) were regressed against time (min; × axis) of sample extraction (i.e., 0, 20, 40, and 60 min). The slope of the resulting best-fit line was then multiplied by the calculated chamber volume (L) and divided by the inner surface area of the chamber (m^2) resulting in flux units of μL CH_4 m^{-2} min^{-1} (Parkin and

Venterea, 2010). The resulting units of the μL CH_4 were then converted using the Ideal Gas Law (PV = nRT) to μmol CH_4, where P was the measured pressure over the 60-min sampling interval in atmospheres (atm), V was the calculated volume of the interior of the chamber (L), n was the number of moles of the gas, R was the gas constant (0.8206 L atm Mol^{-1} K^{-1}), and T was the average measured temperature inside the chamber in Kelvin over the 60-min interval. To convert μmol CH_4 to the mass of CH_4, the molar mass of CH_4 was then used for a final flux unit of mg CH_4 m^{-2} d^{-1} (Parkin and Venterea, 2010). Season-long emissions were calculated on a chamber-by-chamber basis by linear interpolation among measured fluxes between sample dates.

2.6. Plant sampling and processing

Eight days after the last gas sampling (6 September 2016), all aboveground biomass was collected from the interior of each base collar. Plants were cut ~2 cm above the soil surface and dried at 55 °C for 3 weeks then weighed to determine aboveground dry matter. Yield from the field plot was determined using a research-grade plot combine, at which time a sub-sample of rice grain was obtained to determine harvest grain moisture. The combine yield was corrected to 12% grain moisture for yield-reporting purposes. To obtain grain yields from the bins, the panicles were removed from the aboveground dry matter samples from the bins, manually threshed to separate the grain from the panicles, and weighed. Yield was calculated based on grain mass per collar area. Rice grain yields from the bins were corrected to 12% grain moisture. Total season-long CH_4 emissions were divided by total rice grain yield on a bin-by-bin basis to express emissions on a per-unit-grain-yield basis, which has been used as an emissions intensity metric.

2.7. Statistical analyses

Based on a completely random design with three replications of each treatment combination, a single-factor analysis of variance (ANOVA) was conducted using SAS (version 9.4, SAS Institute, Inc., Cary, NC) to determine the pre-assigned treatment (i.e., soil) effects on initial soil properties (i.e., bulk density; sand, silt, and clay fractions; Mehlich-3 extractable P, K, Ca, Mg, S, Na, Fe, Mn, Zn, and Cu contents; soil pH and EC; SOM, TC, and TN contents; and C:N ratio) prior to flooding. A separate two-factor ANOVA was conducted using SAS to determine the effects of soil treatment and time (i.e., measurement date), and their interactions on CH_4 fluxes, soil temperature, and soil Eh. A separate single-factor ANOVA was conducted using SAS to determine the effect of soil treatment on rice yield and season-long area- and yield-scaled CH_4 emissions. All ANOVAs were conducted using the PROC MIXED procedure. When appropriate, means were separated by least significant difference (LSD) at the 0.05 level. Correlation and regression analyses were performed among TC, SOM, sand, silt, and clay and area- and yield-scaled CH_4 emissions using Minitab (ver. 13.31, Minitab Inc., State College, PA).

3. Results and discussion

3.1. Initial soil properties

With the exception of soil EC, all other initial soil properties in the top 10 cm prior to flooding differed ($P < 0.02$) among soil treatments (Table 2). Soil particle-size distributions in the top 10 cm ranged from 0.13 to 0.24 g g^{-1} for sand, from 0.67 to 0.73 g g^{-1} for silt, and from 0.07 to 0.13 g g^{-1} for clay, where each differed ($P \leq 0.01$) somewhat among soil treatments (Table 2). However, particle-size analyses confirmed all soil treatments had a silt-loam texture. Soil pH was mostly alkaline (pH = 7.3) in the 41- and 59-yr-old conventionally tillage agricultural soils, which did not differ, presumably due to the longest history of periodic liming and was mostly acidic (pH = 4.8) in the managed grassland and native prairie soils, which did not differ, presumably due the longest period of undisturbed weathering (Table 2). The recommended soil pH for rice production is between 6.0 and 6.5 (Norman et al., 2013). However, no pH adjustments were made to any soil treatment. Bulk density was similar and largest across all five cultivated agricultural soils put in bins, which averaged 1.46 g cm^{-3}, and was smallest (1.21 g cm^{-3}) in the managed grassland soil (Table 2). Soil organic matter content was greatest in the managed grassland soil (51.0 Mg ha^{-1}) and lowest in the 30-, 41-, and 59-yr-old cultivated agricultural soils placed in bins and the field-plot soil, which did not differ and averaged 24.5 Mg ha^{-1}. Total C, which was considered all organic C, was greatest in the managed grassland soil (24.8 Mg ha^{-1}) and lowest in the 30-yr-old cultivated agricultural soil placed in bins (7.2 Mg ha^{-1}; Table 2). Mehlich-3 extractable soil nutrients ranged from a low of 2.1 times different for K to 5.1 times different for Mg across

Table 2
Analysis of variance summary of the effect of soil treatments on initial soil physical and chemical properties from the top 10 cm prior to flood establishment at the Rice Research and Extension Center near Stuttgart, AR during the 2016 growing season. Means are based on three replications, with the exception of the in-situ field-plot (FP) soil that had four replications.

Soil Property	P	NP[a]	MG	CRP	CA-PT	CA-25	CA-30	CA-41	CA-59	FP
Sand (g g^{-1})	<0.001	0.24a[b]	0.22abc	0.15d	0.13d	0.23ab	0.20bc	0.23a	0.23a	0.20c
Silt (g g^{-1})	<0.001	0.69c	0.67c	0.73ab	0.73a	0.68c	0.68c	0.70bc	0.67c	0.73a
Clay (g g^{-1})	0.01	0.07c	0.11ab	0.12ab	0.13a	0.09bc	0.12ab	0.07c	0.10bc	0.08c
pH	<0.001	4.8e	4.8e	5.6d	6.0c	6.1c	6.8b	7.3a	7.3a	6.9b
Bulk density (g cm^{-3})	<0.001	1.28d	1.21e	1.40bc	1.44ab	1.44ab	1.48a	1.48a	1.47a	1.37c
Extractable nutrients (kg ha^{-1})										
P	<0.001	33.2def	35.1def	19.9f	27.6ef	39.1cde	45.8c	78.5a	41.7 cd	68.9b
K	<0.001	67.5 g	87.3 fg	98.3de	84.1ef	140.9a	117.8bc	104.8 cd	86.0ef	136.0ab
Ca	<0.001	417.8f	1,063e	1,002d	1,361b	1,163c	1,343b	1,614a	1,370b	1,450b
Mg	<0.001	67.3 g	168.7e	214.2c	341.4a	183.3d	139.7e	233.1b	196.0c	121.0f
S	<0.001	47.3b	52.8ab	21.4d	19.9d	30.2c	15.9d	51.0a	30.6c	15.1d
Na	<0.001	21.2e	30.2de	25.4de	41.3c	27.5de	103.5a	51.7b	36.0 cd	67.4b
Fe	<0.001	210.5d	241.8d	230.8d	416.7b	341.5c	312.5c	501.5a	338.8c	489.8ab
Mn	<0.001	234.7d	296.7bc	289.4a	265.7ab	127.2e	233.8bc	138.1e	203.3d	231.5 cd
Zn	<0.001	1.1d	4.9c	1.4d	2.6 cd	4.4c	6.6ab	4.5bc	4.6bc	8.2a
Cu	0.008	1.2d	1.8bc	1.4 cd	2.1a	1.7abc	1.7abc	1.5bc	1.8ab	1.7bc
Total N (Mg ha^{-1})	<0.001	1.57b	2.3a	1.0c	0.9 cd	1.1c	0.6d	1.0c	0.83 cd	0.66d
Total C (Mg ha^{-1})	<0.001	23.7b	24.8a	14.4c	12.5d	13.8 cd	7.2 g	13.2d	10.91e	8.7f
C:N ratio	<0.001	15.1a	10.7d	14.9a	14.4ab	13.2bc	12.0 cd	13.3bc	13.2bc	13.2bc

[a] Treatment abbreviations: native prairie (NP), managed grassland (MG), conservation resource program (CRP), cultivated agriculture (CA-25), cultivated agriculture (CA-30), cultivated agriculture (CA-41), cultivated agriculture (CA-59), and field plot (FP).
[b] Means in same row followed by different letters are significantly different ($P < 0.05$).

all soil treatments (Table 2). Soil K tested low for the 25-yr-old cultivated agriculture and field-plot soil (140 and 136 kg ha^{-1}, respectively) with all other soil treatments in the very low category, with the lowest soil-test K from the native prairie at 67.5 kg ha^{-1}. Soil P was in the optimum range for rice production for the 41-yr-old cultivated agriculture (78.5 kg ha^{-1}), in the medium range for cultivated agriculture CA-25 and CA-59 and in the low soil P range for all other soil treatments, with the lowest of 19.9 kg ha^{-1} for the CRP soil (Norman et al., 2013). No soil amendments were added to correct for any deficiencies. Soil EC averaged 292 dS m^{-1} across all soil treatments. The measured differences among soil treatments were expected, as soils were specifically chosen from various locations and under various landuses to establish a SOM and/or TC gradient for evaluation of SOM/TC concentration effects on CH_4 fluxes and emissions.

3.2. Methane fluxes

During the 2016 rice growing season, CH_4 fluxes followed a predictable temporal pattern, which was similar to previous observations from silt-loam and clay soils in Arkansas (Brye et al., 2013; Rogers et al., 2014; Linquist et al., 2015; Smartt et al., 2016a,b). Methane fluxes started low, increased to numeric peaks that ranged from 232 to 3815 mg CH_4-C m^{-2} d^{-1} between 39 and 53 DAF, which was approximately 50% heading, for the field-plot and native prairie soil, respectively, and decreased thereafter until the flood was released at 75 DAF (Fig. 1). After flood release, CH_4 fluxes in all treatment combinations at least slightly numerically increased within 6 days before decreasing to near zero by 81 DAF (Fig. 1). The occurrence of a post-flood-release increase in CH_4 fluxes has been measured numerous times in both clay and silt-loam soils in Arkansas (Brye et al., 2013; Rogers et al., 2014; Linquist et al., 2015; Adviento-Borbe and Linquist, 2016; Smartt et al., 2016a,b).

During the 2016 growing season, CH_4 fluxes differed ($P < 0.01$; Table 3) among soil treatments over time (Fig. 1). Due to large overall measured variability among all soil treatments, CH_4 fluxes from the CA-30 soil treatment did not differ from a flux of zero on any measurement date throughout the entire growing season. Methane fluxes measured at 5 DAF from all soil treatments did not differ from a flux of zero. At 12 DAF, five soil treatments did not different from a flux of zero (CA-PT, CA-25, CA-41, CA-59, and CA-30), while CH_4 fluxes from the CRP, NP, and MG soil were all greater than a flux of zero. The mean CH_4 flux was larger from the MG than from the NP and CRP soils, which did not differ, at 12 DAF (Fig. 1), presumably due to their large concentration of readily reducible C. At 19 DAF, CH_4 fluxes from four soil treatments (CA-PT, CA-41, CA-59, and CA-30) did not differ from a flux of zero; however, the NP soil had a larger CH_4 flux than the MG soil, while CH_4 fluxes from both the NP and MG soils were greater than fluxes from the CRP and CA-25 soil treatments, which did not differ. At 27 and 32 DAF, CH_4 fluxes from three treatments (CA-41, CA-59, and CA-30) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: NP > MG > CRP > CA-25 = CA-PT. At 39 DAF, CH_4 fluxes from two treatments (CA-59 and CA-30) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: NP > MG > CRP > CA-PT = CA-25 > CA-41. At 47 DAF, CH_4 fluxes from one treatment (CA-30) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: NP > MG > CRP = CA-PT = CA-25 > CA-41 = CA-59.

Between 39 and 53 DAF, CH_4 fluxes numerically peaked for all soil treatments, with the largest numeric peak flux from the NP (3815 mg CH_4-C m^{-2} d^{-1}) and the smallest numeric peak flux from the CA-59 (352 mg CH_4-C m^{-2} d^{-1}) soil at 47 DAF. The CH_4 flux from the CA-59 soil treatment was similar to that reported by Rogers et al. (2014), where CH_4 fluxes from the hybrid CLXL745 ranged from 199 to 448 mg CH_4-C m^{-2} d^{-1}. The peak fluxes reported by Rogers et al. (2014) were substantially lower than the peak fluxes measured from the NP or the MG (2730 mg CH_4-C m^{-2} d^{-1}) treatments, which were five to seven times more than peak fluxes reported by Rogers et al. (2013) (542 mg CH_4-C m^{-2} d^{-1}) on a similar silt-loam soil under Arkansas rice production practices. This dramatic difference in peak CH_4 fluxes is likely due to the native prairie and managed grassland never being under cultivation and their greater concentration of readily reducible C substrate. At 53 DAF, CH_4 fluxes from three treatments (CA-59, CA-41, and CA-30) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: NP > MG > CRP = CA-PT = CA-25. At 61 and 74 DAF, CH_4 fluxes from three treatments (CA-59, CA-41, and CA-30) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: NP = MG > CRP = CA-PT = CA-25. At 76 DAF, which was one day after flood release and similar to 51 and 74 DAF, CH_4 fluxes from three treatments (CA-59, CA-41, and

Table 3
Analysis of variance summary of the effects of soil treatment and time and their interactions on methane fluxes, area-scaled emissions, yield scaled emissions, soil oxidation-reduction (redox) potential, rice dry matter, and rice yield during the 2016 growing season at the Rice Research and Extension Center near Stuttgart, Arkansas.

Variable/source of variation	P
Methane fluxes	
Treatment	<0.01
Time	<0.01
Treatment × time	<0.01
Season-long, area-scaled emissions	
Treatment	<0.01
Soil redox	
Treatment	<0.01
Time	<0.01
Treatment × time	0.99
Rice dry matter	
Treatment	<0.01
Rice yield	
Treatment	<0.01

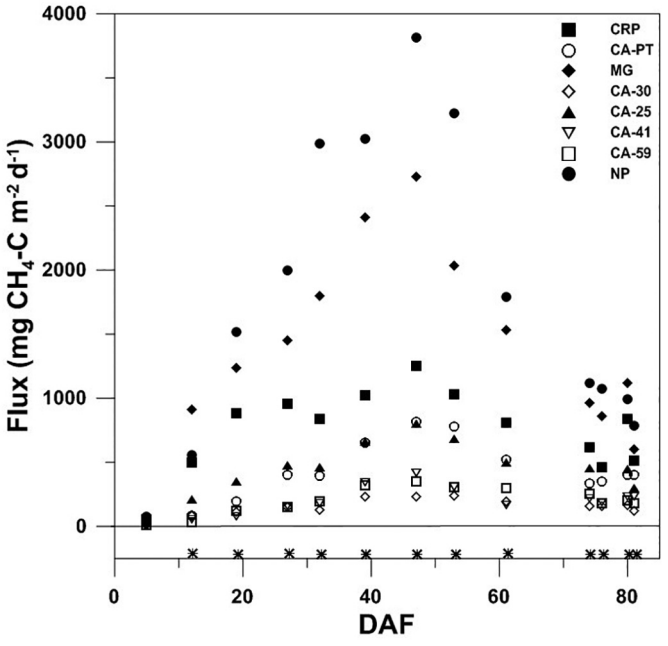

Fig. 1. Season-long profile of methane (CH_4) flux trends over time [i.e., days after flooding (DAF)] for eight soil treatments: native prairie (NP), managed grassland (MG), conservation resource program (CRP), cultivated agriculture (CA-PT) cultivated agriculture (CA-25), cultivated agriculture (CA-30), cultivated agriculture (CA-41), and cultivated agriculture (CA-59), during the 2016 rice growing season at the Rice Research and Extension Center near Stuttgart, AR. A single asterisk (*) on a given measurement date indicates a significant ($P < 0.05$) difference exists from a flux of zero and among soil treatments.

CA-30) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: NP = MG > CRP = CA-PT = CA-25. At 80 DAF, five days after flood release, CH_4 fluxes from the same three treatments (CA-59, CA-41, and CA-30) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: MG = NP and NP = CRP > CA-PT = CA-25. At 81 DAF, six days after flood release, CH_4 fluxes from four treatments (CA-59, CA-41, CA-30, and CA-25) did not differ from a flux of zero, while CH_4 fluxes from the remaining soil treatments ranked, from greatest to smallest, as follows: NP > MG > CRP > CA-PT. The numeric, post-flood-release increases in CH_4 fluxes were consistent with previous observations (Brye et al., 2013; Rogers et al., 2014; Linquist et al., 2015), however not all fluxes returned to zero by the end of the sampling six days after flood release on 81 DAF due to the bins not being fully drained of water.

3.3. Soil temperature and redox potential fluctuations

At the time of flood establishment, soil temperatures at the 7-cm depth averaged 25 °C across all treatments, then increased to the growing-season maximum of 28.6 °C by 6 weeks after flooding (39 DAF), and remained relatively uniform thereafter (Fig. 2). The soil temperature variations measured in this study were similar to those reported by Rogers et al. (2013), where a maximum, 7-cm soil temperature of 32 °C occurred at 19 DAF.

As expected, soil Eh started well-oxidized and decreased ($P < 0.05$) thereafter following flood establishment (Fig. 2). Averaged over soil treatments, soil Eh steadily declined to 39 DAF, then stabilized at around −200 mV. In a similar study on a silt-loam soil in east-central Arkansas, Rogers et al. (2013) reported soil Eh rapidly decreased to < −200 mV by 25 to 30 DAF in a full-season-flood treatment, where a soil Eh of approximately −200 mV is necessary for maximum CH_4 production (Reddy and DeLaune, 2008). Averaged across time, soil Eh was greater (i.e., more oxidized; $P < 0.01$) in the CA-25 soil (131 mV) than that in the CA-41, CA-30, CRP, and NP soils, which did not differ and averaged −169.5 mV (Table 4). The soil treatment differences in Eh correspond well to the measures differences in CH_4 flux trends and peak-flux rankings (Fig. 1).

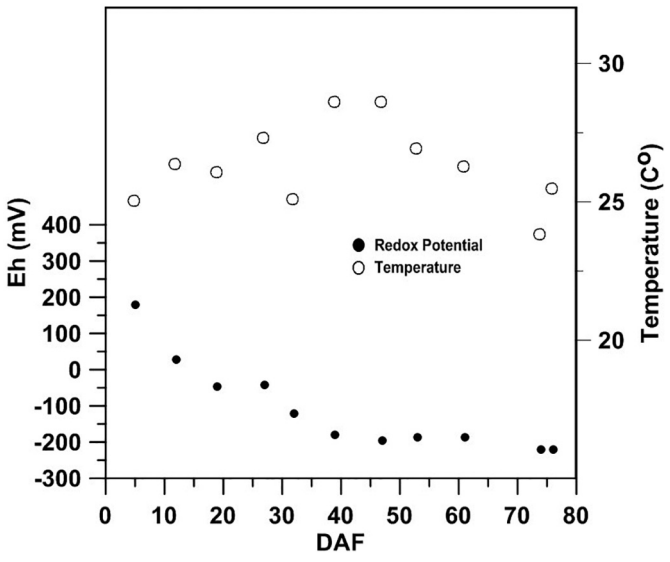

Fig. 2. Season-long profile of soil temperature and oxidation-reduction potential (Eh), measured at the 7-cm soil depth, averaged over time [i.e., days after flooding (DAF)] during the 2016 season at the Rice Research and Extension Center near Stuttgart, AR.

3.4. Area-scaled methane emissions

During the 2016 growing season, season-long, area-scaled emissions ranged from 63 kg CH_4-C ha^{-1} from the in-situ field-plot soil to 1521 kg CH_4-C ha^{-1} from the native prairie soil contained in the bins (Table 4). This 24-fold difference in season-long, area-scaled emission is likely the result of a larger SOM pool in the native prairie soil and/or from the disturbance of the soil from transporting. Similar to that hypothesized, season-long CH_4 emissions differed among soil treatments ($P < 0.01$; Table 3). Season-long CH_4 emissions were greatest from the non-agricultural soils, which also had the greatest SOM and total C contents in the top 10 cm, compared to the current agricultural soils. Season-long CH_4 emissions were 1.3 times greater from the native prairie (1521 kg CH_4-C ha^{-1}) than that from the managed grassland soil (1166 kg CH_4-C ha^{-1}), while both of which were greater than all other agricultural soils (Table 4). The large CH_4 emissions from these two soils (NP and MG) was likely the result of larger SOM contents (45.8 and 51.0 Mg ha^{-1}, respectively) in both soils compared to the other soils for which the lowest average SOM content was 22.1 Mg ha^{-1} for the in-situ field plot soil treatment. Season-long emissions from the managed grassland were 1.9 times greater than that from the CRP soil. Season-long CH_4 emissions from the CRP were 1.8 times greater than that from the cultivated agricultural soil from PTRS (CA-PT) and the 30-yr-old cultivated agricultural soil (CA-30), which did not differ and averaged 352 kg CH_4-C ha^{-1}. Season-long CH_4 emissions from the CA-PT and the CA-30 soils were 2.4 times greater than that from the 25- (CA-25), 41- (CA-41), and 59-yr-old cultivated agricultural (CA-59) soils, which did not differ and averaged 148 kg CH_4-C ha^{-1}. Season-long emissions were lowest among all treatments from the in-situ field-plot soil (Table 4), which also had low SOM and total C contents in the top 10 cm (Table 2). Season-long CH_4 emissions results generally support the expected variations in CH_4 emissions from the differences in SOM contents, where the larger the SOM content in the top 10 cm, the greater the season-long CH_4 emissions.

Similar to the in-situ field-plot and the 30-yr-old cultivated agricultural soil (CA-30) evaluated in this study, Simmonds et al. (2015) reported season-long, area-scaled CH_4 emissions from a silt-loam soil near Stuttgart, AR of 56, 77, 72, and 75 kg CH_4-C ha^{-1} $season^{-1}$ from the hybrid cultivar 'CLXL745' and pure-line cultivars 'Francis', 'Jupiter', and 'Sabine', respectively, grown under a continuous, full-season flood. In addition, Rogers et al. (2013) reported total season-long, area-scaled CH_4 emissions from a full-season-flood on a silt-loam similar near Stuttgart, AR ranged from 54 kg CH_4-C ha^{-1} $season^{-1}$ from N-fertilized bare soil to 220 kg CH_4-C ha^{-1} $season^{-1}$ from the optimally N-fertilized, pure-line cultivar 'Wells'. The substantially larger CH_4 emissions measured in the current study from the native prairie and the managed grassland were likely due to the increased amount of readily reducible C substrate compared to the other soil treatments. With a greater amount of readily reducible amount of C substrate there is a greater potential for increased CH_4 emissions given that a soil C electron acceptor is needed for CH_4 production (Ferry, 1992). There are no other studies that report large CH_4 emissions, such as those measured in the current study from the managed grassland and native prairie. Multiple CH_4 emissions studies in the US (Brye et al., 2013; Rogers et al., 2013; Brye et al., 2016; Smartt et al., 2016a,b) do not report CH_4 emissions within the same order of magnitude as what was measured in this study with regards to the native prairie (1521 kg CH_4-C ha^{-1}) or the managed grassland (1166 kg CH_4-C ha^{-1}). In addition, when taking into account international studies, CH_4 fluxes from deep-water rice in Thailand were lower and averaged ~ 99 kg CH_4 ha^{-1} $season^{-1}$ and rain-fed systems averaged 52 to 91 kg CH_4 ha^{-1} $season^{-1}$ for wet and dry seasons, respectively (Wassman et al., 2000).

Season-long emissions from the 30-yr-old cultivated agricultural soil placed in the bins was two-fold greater than that from the same in-situ soil left in field plots. The difference in emissions between these two treatments was likely due to the soil disturbance that occurred while

Table 4
Summary of soil treatment effects on soil oxidation-reduction (redox) potential of dates sampled, rice dry matter and yield, season-long, area-scaled methane (CH_4) emissions, and CH_4 emissions intensity during the 2016 rice growing season at the Rice research and Extension Center near Stuttgart, AR. Means in a column followed by the same letter do not differ ($P > 0.05$).

Treatment	Redox potential (mV)	Methane emission (kg CH_4-C ha^{-1} season	Dry matter (Mg ha^{-1})	Rice yield (kg ha^{-1})	Emissions intensity [kg CH_4-C (Mg grain)$^{-1}$]
Native prairie (NP)	−137bcd	1521a	35.3ab	16319a	93.7a
Managed grassland (MG)	−121bc	1166b	38.5a	17489a	66.9b
Conservation Reserve Program (CRP)	−149 cd	623c	34.3ab	15447ab	41c
Cultivated agriculture (CA-PT)	−77b	336d	28.9 cd	12099bc	28d
Cultivated agriculture (CA-25)	131a	126e	23.5fe	9374c	13.9ef
Cultivated agriculture (CA-30)	−197d	368d	30.5bc	14578ab	24.9de
Cultivated agriculture (CA-41)	−195d	153e	28.6cde	12786bc	12.1f
Cultivated agriculture (CA-59)	−122bc	166e	24.2def	9698c	17.4def
Cultivated agriculture (FP)	–	63f	23.2f	11289c	5.6f

preparing the bins, where the additional disturbance was apparently enough of a perturbation to the soil structure and aggregation to result in more readily reducible soil C in the bins and/or the existing microbial community was differentially stimulated upon the disturbance to prepare the bins. Consequently, it is likely that the season-long emissions measured from all soil treatments placed in bins were artificially elevated, potentially by a factor of two, compared to what might be expected from the same soil that was left in-situ, cultivated, and cropped to rice under an optimally N-fertilized, full-season flood management system. Regardless of the potential over-estimation of season-long CH_4 emissions, results of this study clearly demonstrate a relationship exists between season-long CH_4 emissions and initial SOM and/or soil C that supplies reducible C substrate for methanogenesis.

3.5. Rice dry matter and yields

Rice dry matter ranged from 23.2 to 38.5 Mg ha^{-1} from the in-situ field-plot and managed grassland soil, respectively, while rice yields ranged from 9.7 to 17.5 Mg ha^{-1} from the 25-yr-old cultivated agricultural soil, which was the disturbed-soil counterpart to the in-situ soil from the traditional field plots, and managed grassland soil, respectively (Table 4). The large yield produced from the managed grassland soil was out of the range of typical plot-scale yields (Hardke et al., 2016). However, the large rice yield also demonstrates the potential substantial influence that an undisturbed soil can have on plant productivity due to the inherent natural soil fertility associated with non-cultivated grassland soils. Despite the same rice variety being planted and grown in the field plots and in all prepared soil bins, rice dry matter ($P < 0.01$) and rice yields differed ($P < 0.01$) among soil treatments (Table 3). Rice dry matter was >25% greater from the managed grassland than from all other currently cultivated agricultural soils. Rice dry matter was the lowest from the in-situ field-plot soil. Similar to dry matter, rice yield was >35% greater from the managed grassland than from the 25-, 41-, and 59-yr-old and PT cultivated agricultural and the in-situ field-plot soil. For comparison, based on Arkansas Rice Performance Trials in 2016, yields for rice cultivar 'LaKast' grown under a continuous, full-season-flood regime on a Dewitt silt-loam soil near Stuttgart, AR averaged 9.5 Mg ha^{-1} (Hardke et al., 2016). Consequently, rice growth and productivity from field plots and bins in this study performed reasonably similar to production-scale rice productivity. In contrast to season-long CH_4 emissions, neither rice dry matter nor yield differed between the in-situ field-plot soil and the same soil placed in the prepared bins (Table 4), suggesting that plant growth was similar when rice was grown in the prepared soil bins compared to rice grown under typical conditions in in-situ field-plot soil. Though season-long CH_4 emissions were two times greater from the soil placed in the bins compared to the in-situ field-plot soil, plant-response results demonstrated that preparing small-scale bins to evaluate CH_4 emissions from widely differing soils from various sites at a single location to impose uniform management was a reasonable approach.

3.6. Methane emissions intensity

Improving CH_4 emissions intensity by reducing CH_4 emissions per unit grain yield produced should be a management goal for rice producers to maintain sustainable rice production and resource use into the future. For the 2016 rice growing season, CH_4 emissions intensity ranged from 5.6 kg CH_4-C (Mg grain)$^{-1}$ in the in-situ field-plot soil to 93.7 kg CH_4-C (Mg grain)$^{-1}$ in the native prairie soil (Table 4). Consequently, as suspected, CH_4 emissions intensity differed ($P < 0.01$; Table 4) among soil treatments. Emissions intensity, where, based on how the calculation was conducted, the larger the value, the larger the intensity, from the native prairie was greater from than that from the managed grassland, which was greater than from the CRP soil (Table 4). Emission intensity was lowest from the 25-, 41-, and 59-yr-old cultivated agricultural soils placed in bins and the in-situ field-plot soil, which did not differ and averaged 12.2 kg CH_4-C (Mg grain)$^{-1}$.

Emissions intensity results from this study for the cultivated agricultural soils placed in bins and for the in-situ field-plot soil were comparable to those from Rogers et al. (2013) and Simmonds et al. (2015), who both measured CH_4 emissions from rice grown under a continuous, full-season flood on a silt-loam soil near Stuttgart, AR. Rogers et al. (2013) reported a CH_4 emissions intensity of 27.6 kg CH_4-C (Mg grain)$^{-1}$ with the pure-line cultivar 'Wells', whereas Simmonds et al. (2015) reported an average CH_4 emissions intensity of 6.8 kg CH_4-C Mg grain^{-1} for the hybrid cultivar 'CLXP4534' and the pure-line cultivars 'Francis', 'Jupiter', and 'Sabine' averaged 10.9 kg CH_4-C Mg grain^{-1}. The differences in emissions intensity can be attributed to yield differences in the Simmonds et al. (2015) field study, whereas yields were greater for the current field study for the lowest-emitting soil treatments, thus improving/reducing the emissions intensity. The largest-yielding soil treatments in the current study also had CH_4 emissions that were orders of magnitude larger than what has been documented in any study thus far on a silt-loam soil, which greatly increased/worsened the emissions intensity.

3.7. Relationship between SOM, C, and CH_4 emissions

Several measured soil properties in the top 10 cm correlated with CH_4 emissions (Tables 5 and 6). Both SOM and TC concentrations were strongly, positively correlated ($r > 0.86$; $P < 0.01$) with season-long CH_4 emissions and emissions intensity (Table 5). However, both SOM and TC contents were unrelated to season-long CH_4 emissions or emissions intensity likely due to the added variation in estimated bulk density as part of the content calculation, but potassium ($r = -0.41$, $P = .05$) and zinc ($r = -0.46$, $P = .01$) contents were both moderately negatively correlated season-long CH_4 emissions and emissions intensity (Table 6). Similar to that hypothesized, the result of the correlations indicates that both CH_4 emissions and emissions intensity increase as SOM or TC concentration increase, which further validates the initial goal of the selected soils representing a gradient of SOM/TC

Table 5
Linear correlation summary among measured soil properties and concentrations in the top 10 cm and season-long, area-scaled methane (CH_4) emissions and emissions intensity [kg CH_4-C (Mg grain)$^{-1}$] for the 2016 growing season at the Rice Research and Extension Center near Stuttgart, AR. Single asterisk (*) indicates P <0.001.

Soil property	Season-long CH_4 emissions	Emissions intensity
	r	
Sand (g g^{-1})	0.159	0.109
Silt (g g^{-1})	−0.019	0.018
Clay (g g^{-1})	−0.214	−0.188
Soil organic matter (g kg^{-1})	0.899*	0.860*
Total carbon (g kg^{-1})	0.924*	0.884*

concentration for CH_4 emissions evaluation in this study. However, in laboratory studies in Louisiana, using 16 soils ranging in texture from silt to clay, CH_4 emissions and SOM concentrations in the range of 14 to 23.8 g kg^{-1} were examined (Wang et al., 1993). Wang et al. (1993) reported that no correlation existed between CH_4 emissions and soil properties such as nitrogen, pH, or cation exchange capacity, but there was a significant increase in CH_4 entrapment in soils with large clay contents, which suggested that soil texture plays a vital role in CH_4 emissions (Wang et al., 1993). In a study based in Japan, no correlation was reported between total CH_4 emissions and any single measured soil property, which included amorphous Fe(III), free iron (Fe)(III), easily reducible manganese (Mn), nitrate (NO^{3-}), and sulfate (SO_4^{2-}), and reducing agents including total (C), total nitrogen (N), and easily decomposable C (Watanabe and Kimura, 1999).

Soil particle-size fractions were unrelated ($P > 0.05$) to season-long CH_4 emissions and emissions intensity, likely due to narrow ranges since only silt-loam soils were targeted for evaluation in this study. Other studies have indicated that there are greater CH_4 emissions from silt-loam than from more clayey soils (Brye et al., 2013; Smartt et al., 2016b). With the large and significant linear correlations, the possibility exists to estimate season-long CH_4 emissions and emissions intensity with some confidence from only a few basic measured soil properties from the top 10 cm. Both SOM ($R^2 = 0.81$; $P < 0.01$) and TC ($R^2 = 0.85$; $P < 0.01$) concentrations from the top 10 cm produced strong, positive linear relationships with season-long, area-scaled CH_4 emissions (Fig. 3). Similar studies from Brye et al. (2013) and Rogers et al. (2014) have produced season-long CH_4 emissions of 159.6 and 190 kg CH_4-C ha^{-1}, respectively, with corresponding mean TC contents of 8.7 and 11.4 Mg ha^{-1}, respectively.

3.8. Agronomic and environmental implications

This field experiment provided an opportunity to examine the relationship between CH_4 production and initial SOM and/or TC contents. Results clearly demonstrated that initial SOM/TC content affects CH_4 fluxes and season-long emissions, specifically in that both CH_4 fluxes and season-long emissions tended to be greater when the initial SOM/TC contents were large. The demonstrated relationship has ramifications for SOM conservation, soil C sequestration, climate change, and soil health with regards to rice production in the Lower Mississippi River Valley of the US.

Table 6
Linear correlations summary among measured soil property contents in the top 10 cm and season-long, area-scaled methane (CH_4) emissions and emissions intensity [kg CH_4-C (Mg grain)$^{-1}$] for the 2016 growing season at the Rice Research and Extension Center near Stuttgart, AR. Single asterisk (*) indicates P < 0.05.

Soil property	Season-long CH_4 emissions	Emissions intensity
	r	
Zinc (kg ha^{-1})	−0.495*	−0.504*
Potassium (kg ha^{-1})	−0.407*	−0.439*
Total carbon (Mg ha^{-1})	0.338	0.344

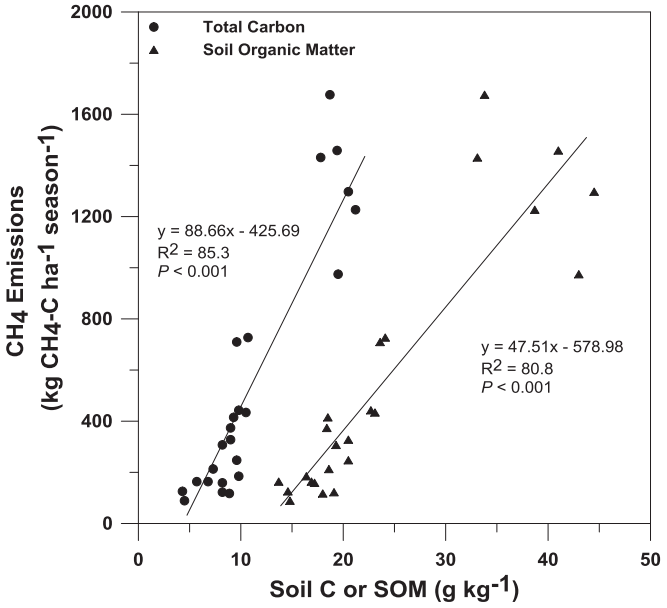

Fig. 3. Relationship between soil organic matter (SOM) and total carbon (TC) concentration in the top 10 cm and season-long, area-scaled methane (CH_4) emissions from the 2016 rice growing season at the Rice Research and Extension Center near Stuttgart, AR.

The general decrease in SOM under rice production from extensive tillage also decreases soil's native C and decreases soil tilth and structure, and overall health, which in turn can result in decreased soil fertility (Mosier et al., 2004). Native, or undisturbed, soils often act as a C sink, sequestering large quantities of C through natural processes (Mosier et al., 2004). When those soils are disturbed, that sink can transforms into a tremendous C source by way of anaerobic and aerobic microbial activity, adding to the C load to the atmosphere and increasing GHG concentrations. Utilizing soils that have increased SOM/TC for rice production will likely increase CH_4 emissions, as shown with the current research, which may contribute to potential negative effects of global climate change. If rice production is expanded into previously uncultivated land areas, the environmental impact with respect to GHG emissions may be more severe than previously thought. Though it is still unclear due to the lack of research, the full range of potential/expected GHG emissions following the conversion of previously minimally managed, non-agricultural land to intensively managed, row-crop production, the results of the current study provide an initial baseline for what could potentially occur with regards to CH_4 emissions in the short-term following land-use change. As humans become more interested in long-term sustainability and seek ways to mitigate the sources and impacts of climate change, this research can serve as piece of the framework to help determine the C budget needed to reduce the effects of agriculture on climate change.

4. Conclusions

To the author's knowledge, this was the first field experiment to select, transport, and combine multiple soil treatments from various locations into a single study at one location so that production practices (i.e., planted rice cultivar, N fertilization, water management) and environmental conditions (i.e., precipitation, air temperature variations) could be uniform among treatments, where the main variable was SOM/TC content. Verifying the hypothesis, results of this study showed that CH_4 emissions and emissions intensity were greatly affected by initial SOM/TC content and confirmed a strong, positive relationship between season-long, area-scaled CH_4 emissions and TC and SOM contents in the top 10 cm. Though season-long CH_4 emissions were greater from soil placed in bins than from in-situ field-plot soil, rice

dry matter and yields were unaffected, indicating that the bin approach used in this field study was a reasonable methodology for evaluating the effects of initial SOM/TC contents on CH_4 fluxes and emissions across a wide range of initial SOM/TC contents from geographically diverse areas.

Based on increased season-long, area-scaled CH_4 emissions from previously non-agriculturally managed soils, conversion to intensive agricultural practices and rice production may have a large climate-change impact, at least in the short-term. Though the results of this study were based on one growing season of measurements, these results indicate relatively consistent CH_4 emissions responses from year to year are at least partially due to the presence of the flood water for most of the growing season attenuating climate variations and inter-annual differences in growing-season weather conditions. Therefore, it is reasonable to conclude that these results can be extrapolated to future years and similar field conditions, as minor differences in growing-season weather conditions from year to year likely have minimal effect on CH_4 emissions. As efforts increase to mitigate climate change globally and public discourse shifts to align itself with environmental stewardship, one way to forward the goal of reduced GHG emissions from rice production is to be acutely aware of all soil and management practice factors that affect GHG emissions and plan agricultural practices accordingly to minimize the C footprint and maximize future resource sustainability. Rice production must attain a level of sustainability that will aid the goal of feeding an ever-growing human population, and GHG emissions are a key part of this modern puzzle. There is a responsibility to maximize production of staple grains, while bearing in mind that humans must equally protect future generations from the devastating effects of global climate change.

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